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# Mitigating farm-scale greenhouse gas emissions by removing cows in response to wet soil conditions.

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## Abstract

In pasture-based grazing systems, urine deposition is the major source of the greenhouse gas nitrous oxide ($N_2O$). Livestock treading damage and high soil water contents increase the risk of $N_2O$ emissions. Duration controlled grazing (DCG) practices that are implemented in response to soil water conditions above a threshold may therefore provide an effective means of reducing greenhouse gas (GHG) emissions from dairy farms. In this study we used the DairyNZ Whole Farm Model and APSIM model to assess the cost-benefit of implementing DCG to reduce total $N_2O$ and manure-derived $CH_4$ emissions from dairy farms. We modelled scenarios on poorly drained or imperfectly drained soils in four regions of New Zealand including Waikato, Manawatu, Canterbury and Southland, where the grazing time on wet days was 0, 13, 17 or 21 hours per day. Emissions were estimated using a refined version of New Zealand's current national greenhouse gas inventory methodology. Our analysis suggested that reducing the grazing time from 21 hours to 0, 13 or 17 hours per day when soils were wet could reduce annual $N_2O$ and manure-derived $CH_4$ emissions by up to, respectively, 12, 9 or 5% on farms with poorly drained soils. The 13 hour per day grazing duration was the least costly, particularly if

there were more than 150 ‘wet’ days per year. In contrast, for dairy farms on imperfectly-drained soils, DCG increased emissions, suggesting this management approach for reducing GHG emissions is not suitable for these soils.

**Keywords**: Modelling, Whole Farm Model, APSIM, nitrous oxide, duration controlled grazing.

## 1 Introduction

Nitrous oxide ($N_2O$) is an important anthropogenic greenhouse gas (GHG), with agriculture its largest source (Reay et al., 2012). About one third of these global emissions are attributed to excreta returns during livestock grazing (Oenema et al., 1997). Grazing livestock excrete 75-90% of their nitrogen (N) intake in concentrated urine and dung patches (Whitehead, 1995). When deposited on land, the urine-N returns, ranging from 200 to 2000 kg N $ha^{-1}$ for cattle (Selbie et al., 2015), exceed plant uptake capacity and can lead to significant N losses through leaching (Ryden et al., 1984) and gaseous N emissions, including $N_2O$ and ammonia ($NH_3$) (de Klein et al., 2001). Both N leaching and $NH_3$ emissions are sources of indirect $N_2O$ emissions (Butterbach-Bahl and Dannenmann, 2011). In New Zealand, ruminant livestock excreta deposition onto pastures is the single largest source of $N_2O$, contributing *c*. 80% of the direct and indirect $N_2O$ emissions (de Klein et al., 2006). Under urine patches, $N_2O$ production and emission will be primarily influenced by oxygen availability which is regulated by soil water content (Linn and Doran, 1984; de Klein et al., 2006). $N_2O$ emission factors have been developed for dairy urine deposited on pasture that incorporate soil water content (van der Weerden et al., 2014). A lower oxygen diffusion rate in soils that have been compacted as a result of animal treading damage can further promote $N_2O$ emissions via denitrification (Ball et al., 2012, van Groenigen et al., 2005).

The New Zealand dairy industry aims to increase milk production and reduce greenhouse gas emissions, and acknowledges the challenge in achieving these, sometimes, opposing objectives (Beukes et al., 2011). One particular farm practice that may achieve both objectives is duration controlled grazing (DCG) during wet periods of the year, whereby cow grazing times are reduced with time spent on off-paddock facilities (e.g. standoff pads) for a part of the day. The reduction in grazing hours reduces the amount of excreta N deposited onto wet soils, thereby reducing direct and indirect (via $NO_3$ leaching) $N_2O$ emissions (de Klein et al., 2006; Christensen et al., 2012; Luo et al., 2013). This practice also protects soils from animal treading damage (Houlbrooke et al., 2009), which in turn may lead to increased pasture production, and, through careful pasture management, can be converted into increased milk production. Measurements reported by de Klein et al. (2006) from southern New Zealand showed that DCG reduced $N_2O$ emissions and $NO_3$ leaching from paddocks by approximately 40% when cows were on pasture for 3 hours per day during March, April and May compared to 21 hours (normal rotational grazing practices, allowing 3 hours for milking per day). Similarly, in northern New Zealand, Luo et al. (2013) observed 55% reduction in $N_2O$ emissions during spring (September and October) when cow grazing hours during winter (June to August) were reduced from 24 to 6 hours per day.

Adoption of DCG practices will increase the volume of excreta that is captured and stored from the off-paddock facility (Luo et al., 2013). Any increase in the volume of excreta stored in manure management systems will increase $N_2O$, $NH_3$ and methane ($CH_4$) emissions from this component of the farm system (Chadwick et al., 2011; Laubach et al., 2015). Therefore, there is potential that DCG practices may lead to ‘pollution swapping’, whereby the emissions from increased manure management potentially over-ride corresponding reductions achieved from avoiding grazing of wet paddocks. Furthermore, the period of time cows are removed from the paddock invariably increases operational costs such as those associated with supplying a quality feed supplement, effluent management and maintenance of the stand-off facilities.

A recent analysis of GHG mitigation options showed that a calendar-based approach (i.e. removing cows every day over a certain timeframe e.g. spring) to using standoff pads was not cost-effective (Adler et al., 2015). In order to meet both economic/production and environmental (avoiding pollution swapping) objectives, it is important that cows are removed from paddocks only when it is necessary to do so.

Ultimately, farmers will be attracted to options that provide on-farm production and/or financial benefits. Therefore, the objective of this study was to investigate whether tactical removal of dairy cattle from wet paddocks could provide a cost-effective option for reducing farm-scale $N_2O$ and manure-derived $CH_4$ emissions. To achieve this objective, we (i) developed a relationship between soil volumetric water content (VWC) and $N_2O$ emissions from urine deposition, (ii) modelled excreta cycling and N losses for typical dairy farms in the Waikato, Manawatu, Canterbury and Southland regions of New Zealand, (iii) employed a refined version of New Zealand's greenhouse gas inventory methodology based on the latest available science, and (iv) assessed the cost:benefit of this approach for reducing greenhouse gas emissions. This final step was achieved by utilising the modelled productivity and economic results of implementing DCG when soils were wet, reported in an associated paper (Laurenson et al., submitted).

# 2 Methodology

## 2.1 Overview of approach

We used a combination of models and existing knowledge to assess the impact of DCG scenarios on $N_2O$ and manure-derived $CH_4$ emissions for case study dairy farms in four regions of New Zealand: Waikato, Manawatu, Canterbury and Southland. For each farm we used the DairyNZ Whole Farm Model (WFM; Beukes et al., 2008) to estimate excreta N deposition for a 'baseline' farm and three scenarios that included varying grazing durations on days when soils were wet (see section 2.2). Modelled excreta N for each farm scenario was used to estimate direct $N_2O$ emissions employing $N_2O$ emission factors based on a relationship between soil VWC and $N_2O$ emissions (section 2.3). The urine N excretion values estimated by the WFM were also used within the Agricultural Production Systems Simulator (APSIM; Holzworth et al., 2014) modelling framework to assess N leaching and $NH_3$ emissions from urine patches and N fertiliser for the different farms and scenarios under three rainfall regimes (section 2.4). Leaching losses from dung deposited in the paddock and manure (solid or liquid) from the off-paddock facility were estimated using WFM modelled N loading rates combined with the N leaching fraction used in the New Zealand $N_2O$ inventory methodology (section 2.5). Manure-derived $CH_4$ emissions were estimated using a combination of the New Zealand IPCC inventory methodology and the default IPCC approach (IPCC, 2006; Ministry for the Environment, 2015). For comparative purposes we also estimated farm-scale $N_2O$ and manure-derived $CH_4$ emissions using emission factors from the NZ GHG inventory methodology (section 2.6). The cost:benefit of the proposed DCG approach was estimated using modelled farm operating profits (Laurenson et al., submitted) and estimated GHG emissions, and is expressed as \$/kg carbon dioxide equivalents ($CO_2e$) reduction achieved through the adoption of DCG (section 2.7).

## 2.2 Modelling excreta N deposition

The DairyNZ WFM was used for estimating excreta N production. This model has been used in New Zealand to model farm management strategies and productivity for a range of pastoral dairy systems (Beukes et al., 2008). A full description of the WFM model can be found in Beukes et al. (2013). In brief, the model framework represents a pasture-based dairy farm with individual paddocks and cows simulated on a daily time step. Cow feed intake is driven by metabolic demand determined by a mechanistic and dynamic model within the WFM that simulates critical elements of cow digestion and metabolism (Hanigan et al., 2009). The cow model predicts daily milksolids production (MS = fat + protein), outputs of N in urine, faeces and milk N output, and methane emissions. The pasture-soil model in WFM (Romera et al., 2009) is climate-driven using daily weather data accessed from the National Institute of Water and Atmospheric Research Virtual Climate Station (VCS) network (Tait et al., 2006).

We determined excreta N deposition by modelling dairy farms in four regions including Waikato, Manawatu, Canterbury and Southland that were located on either poorly drained or imperfectly drained soils (Table 1). We used the same soil characteristics for poorly drained and imperfectly drained soils within each region to allow a comparison of the impact of contrasting regional climates on the effectiveness of DCG to reduce GHG emissions. It is important to note that individual simulated farms did not include combinations of both soil drainage classes. The poorly drained soil, a Temuka clay loam, is classified as a Typic Orthic Gley soil by the New Zealand soil classification (Hewitt, 2010; 47% clay in top 100 mm) or Mollic Endoaquept by USDA soil taxonomy (Soil Survey Staff, 1998). The imperfectly-drained soil, a Hatfield silt loam, is classified as a Typic Immature Pallic (Hewitt, 2010; 20% clay in top 100 mm) or Udic Haplustept (USDA soil taxonomy; Soil Survey Staff, 1998). Cow stocking rate (SR) was set at a level which ensured that the simulated farms were suitably stocked relative to the pasture grown (Table 1). All regions used the same SR for the poorly and imperfectly drained soils, apart from Southland, where the SR for the poorly drained soil was slightly higher (3.15) than for the imperfectly-drained soil (2.75) due to the large difference in typical pasture production across soils in this region (Laurenson et al., submitted).

*Insert Table 1*

Duration controlled grazing was imposed when a field's soil VWC exceeded a critical water content (CWC) at the time of grazing. This CWC was defined as the VWC when the risk of treading damage is at its greatest (Piwowarczyk et al., 2011), and varied with soil drainage class. Cows were removed from paddocks if the VWC was greater than 85% of field capacity (FC) on poorly drained soils and 105% of FC on imperfectly drained soils (Laurenson et al., in prep). We compared the CWC with the modelled soil water balance to estimate how many days per year cows should be removed from paddocks due to a risk of treading damage. On the days when VWC > CWC, grazing time per day was either 0 hours (i.e. complete removal), 13, 17 hours or 21 hours, where 21 hours represented the baseline in which no restriction was placed on grazing duration. The 0, 13 and 17 hours related to, respectively, 21, 8 or 4 hours on an off-paddock facility (standoff pad). The standoff pad was assumed to have a pine bark and sawdust base (Luo et al., 2008) and was located within 250 m of the milking parlour. It was assumed that cows remained on pasture year round in warmer northern regions (Waikato and Manawatu) where winter pasture growth meets feed demand. In the cooler southern regions, non-lactating cows were 'wintered off' farm between 1 June and 8 August, reflecting typical dairy farm practice. Therefore, this analysis considered 365 days of the year in the two northern regions, while the assessment was restricted to the 270 days lactation season (commencing 9 Aug) in the two southern regions.

When DCG was not imposed, animals were either on the paddocks, on a lane or in parlour and yards. The amount of urine-N excreted onto these surfaces was proportional to the time spent on each. Cows spent 1 hour per day on lanes and, during the lactation season, 2 hours per day in the dairy parlour and yards, with the remaining time was spent on paddocks. Outputs from the WFM included the amount of N deposited as dung and urine onto paddocks, dairy parlour and yards, lanes and standoff areas; the volume of effluent collected, stored and applied to the soil; production and economics data from each simulation. The latter model output has been reported in an accompanying paper (Laurenson et al., submitted).

## 2.3 Relationship between soil water content and $N_2O$ emissions

Previous research has shown that $N_2O$ emission factors ($EF_3$ which quantifies the percentage of applied N lost as $N_2O$) for dairy cattle urine are strongly related to the soil water filled pore space (WFPS) averaged over 30 days following urine deposition (van der Weerden et al., 2014). For the current study, we adopted VWC as the soil water metric, as it has the advantage of being relatively easy to determine using field sensors and directly compatible with soil water balances under field conditions (van der Weerden et al., 2012). Using $N_2O$ and soil type data from 31 field trials (collated from de Klein et al., 2003, 2004; Luo et al., 2008; Sherlock et al., 2003a,b; Thomas et al., unpubl. data and van der Weerden et al., 2011) we employed the APSIM model to estimate VWC at various soil depths (75, 150 and 200 mm) and for different number of days following urine deposition (15, 20, 30, 45 and 60 days). We then investigated which depth and number of days produced the strongest relationship between modelled VWC and measured $EF_3$.

## 2.4 Estimating N leaching and $NH_3$ emissions from urine and N fertiliser

As the WFM does not calculate nitrate ($NO_3$) leaching and $NH_3$ emissions, we used the estimated amount of excreta N as input parameters to the APSIM model. In New Zealand APSIM has been validated against a range of drainage and leaching regimes that occur under urine-patch conditions (Cichota et al., 2012; 2013). Pasture growth is simulated using AgPasture (Li et al., 2011), with a ryegrass clover mixture, the SoilN and SurfaceOM modules (Probert et al., 1998) were used to describe the C-N cycle, and SWIM2 for the transport of water and solutes, which is based on the Richards' equation and the convection-dispersion equation and the Micromet module (Snow and Huth, 2004) for computing evapotranspiration and energy partition. Also included was a module accounting for volatilisation from urine patches and N fertiliser based on the approach by Génermont and Cellier (1997).

Monthly values of urine patch N load (kg/ha) per day, as obtained from the WFM, were used in the APSIM modelling framework to generate estimates of N leaching and $NH_3$ emissions. APSIM simulations ran for a two year period following urine deposition to ensure that all leached N was accounted for. Within a given paddock, N leached from the urine patch were aggregated with N leached from non-urine affected area thereby providing a single N leaching value (Vogeler et al., 2013). A similar approach was taken for modelling and aggregating $NH_3$ emissions from urine patches and N fertiliser applications. Annual N fertilisation rates differed between regions and farm scenarios, ranging from 68 to 254 kg N/ha. Fertiliser N rates were reduced to account for any N applied in farm dairy effluent (FDE) collected from the standoff pad and solid manure scraped from the pad. It was assumed 85% and 40% of the total N in FDE and solid manure, respectively, would become available for pasture uptake (Gutser et al., 2005; Webb et al., 2013).

## 2.5 Estimating N leaching and $NH_3$ emissions from dung, effluent and solid manure

Paddock N inputs as dung, solid and liquid manure were estimated using the WFM, with effluent applied as necessary (Laurenson et al. submitted). As APSIM has not been validated for N losses from dung, solid and liquid manure, subsequent N leaching and $NH_3$ emissions were based on the New Zealand $N_2O$ inventory methodology, where it was assumed, respectively, 7% and 10% of N inputs were leached as $NO_3$ and volatilised as $NH_3$ (Ministry for the Environment, 2015).

## 2.6 Farm-scale $N_2O$ and manure-derived $CH_4$ emissions from modelled dairy farms

Direct $N_2O$ emissions from paddocks are reported as kg $N_2O$-N/ha/year, and were calculated using the VWC function (previously described in section 2.3) for determining cattle urine $EF_3$. Total $N_2O$ and manure-derived $CH_4$ emissions were calculated for each dairy farm scenario and reported on the basis of kg $CO_2$e/ha/year, where $N_2O$ and $CH_4$ have global warming potentials of 298 and 25 times that of $CO_2$, respectively, over a 100-

year time horizon, as used by the IPCC (Forster et al., 2007). These total emissions were calculated using a refined version of the New Zealand IPCC inventory methodology (Ministry for the Environment, 2015). Key refinements include (i) $N_2O$ emissions from urine deposited onto paddocks estimated using the relationship developed between VWC and $N_2O$ emission factors, and (ii) improved estimation of $NH_3$ and $NO_3^-$ losses from urine and urea fertiliser using a modelling approach (APSIM); Table 2 lists all refinements and assumptions employed. We also categorised all excreta deposited onto standoff pads as 'solid storage', based on the definitions of manure management systems (Table 10.18, IPCC 2006). IPCC default values were employed except for direct $N_2O$ emissions from solid storage ($EF_{3\,S}$), where we used results from a New Zealand study ($EF_{3\,S}$ = 0.01%; Luo and Saggar, 2008). We also assumed 4% of total N excreted onto standoff pads drained into FDE ponds, based on research by Luo et al. (2008). We estimated $CH_4$ emissions from standoff pads (kg $CH_4$/cow/year) by assuming volatile solids (VS) were 3.5 kg dry matter/cow/day, corrected for the time on the standoff, maximum $CH_4$ producing capacity for manure from cattle ($B_O$) was 0.24 $m^3$ $CH_4$/kg VS and a $CH_4$ conversion factor (MCF, %) of 4% (equation 10.23, IPCC 2006). Modelling was conducted for three individual years for each region, representing years when rainfall depth was equivalent to the $10^{th}$, $50^{th}$ and $90^{th}$ percentile for years between 1995 and 2014. Presentation and discussion of modelling data focuses primarily on results from the $50^{th}$ percentile rainfall year (20-year average), while data from all modelled years were used when analysing cross-regional relationships.

*Insert Table 2*

We excluded the $CH_4$ emissions from enteric fermentation from all calculations of total greenhouse gas emissions, as the modelled farms maintained the same annual dry matter intake per cow and therefore the same $CH_4$ emissions (Clark et al., 2003) regardless of whether cows remained on, or were removed from, paddocks. We also present farm-scale $N_2O$ and manure-derived $CH_4$ emissions based on the current inventory methodology, as a comparison to the refined approach. Paddock-derived $N_2O$ emissions were estimated using the New Zealand-specific $EF_3$ value of 1% of urine N deposited, as employed in the current New Zealand $N_2O$ inventory. The current inventory methodology does not account for manure collected on standoff pads. Therefore, it was assumed that all off-paddock excreta deposition would be accounted for as effluent stored in anaerobic lagoons, as is currently conducted within the New Zealand agricultural greenhouse gas inventory.

### 2.7 Cost:benefit of DCG

The financial cost or benefit from adopting DCG was calculated from the change in dairy operating profit (Table 3), as determined from the economics component of the WFM (Beukes et al., 2013) and total $N_2O$ and manure-derived $CH_4$ emissions (current study). The dairy operating profit considered the most relevant farm variables (e.g. sale of MS and culled stock, enterprise costs such as insurance, labour expenses and farm system capital and operating costs). The cost-benefit was based on a long term milksolids (MS) price of NZ$6 per kg MS, and is presented as $/kg $CO_2$e reduction achieved through the adoption of DCG.

*Insert Table 3*

## 3 Results

### 3.1 Direct $N_2O$ emissions from urine deposition onto paddocks

Nitrous oxide emissions from pastoral soils increased with soil water content due to anaerobic conditions stimulating denitrification activity. The strongest relationship between soil water content and $EF_3$ was observed when VWC in the top 75 mm of soil was averaged over 20 days following urine deposition ($VWC_{20d}$; $R^2$ = 0.42; P < 0.001; Fig.1). Using this relationship, modelled $N_2O$ emissions from urine deposited onto paddocks ranged from 2.6 to 2.7 kg $N_2O$-N/ha/year from the poorly drained soils in all four regions when cows remained on paddocks (Fig. 2a, 2c, 2e and 2g). When cows were completely removed from wet paddocks, emissions from poorly drained soils in the two South Island regions were reduced by 38-54%, while a reduction of 76-82% was predicted for farms in the two North Island regions. In contrast, $N_2O$ emissions from the imperfectly-drained soil were low when cows remained on wet soils due to the relatively lower VWC, with emissions ranging from 0.54-0.78 kg $N_2O$-N/ha/year. Completely removing cows from paddocks when imperfectly drained soils were wet reduced paddock-derived $N_2O$ emissions by 49-59% in the two North Island regions, whereas a relatively small reduction of 6% was calculated for the South Island farms due to cows wintered off in June and July which reduced the frequency of grazing events that occurred on 'wet' days.

Emissions of $N_2O$ from urine deposition based on the current IPCC methodology are estimated as the product of N load and $EF_3$, where the latter has a value of 1%, regardless of soil water content. Therefore, for the baseline, $N_2O$ emissions from urine deposition were the same for the two soil drainage classes within each region in Waikato, Manawatu and Canterbury since the amount of urine-N deposition (i.e. N load) was the same (Fig. 2b, 2d and 2f). In contrast, Southland showed slightly higher $N_2O$ emissions per hectare for the poorly drained soil when DCG was not implemented (Fig. 2h) due to slightly higher stocking rate at 3.15 cows/ha compared to 2.75 cows/ha for imperfectly drained soils and therefore N load onto the soil (Table 1). Implementing DCG when soils were wet reduced direct $N_2O$ emissions from paddocks in all regions (Fig. 2b, 2d, 2f and 2h), reflecting the lower amount of urine N that was deposited onto pasture and the lower EF value for standoff pads (0.0001; Table 2).

*Insert Figure 1*

*Insert Figure 2*

3.2 Farm-scale $N_2O$ and manure-derived $CH_4$ emissions from dairy farms

We consider the refined inventory methodology provides a more accurate assessment of the impact of our DCG strategy on total $N_2O$ and manure-derived $CH_4$ emissions at the farm-scale. However, we include a comparison with the current New Zealand inventory methodology (section 3.3) to illustrate the difference in total $N_2O$ and manure-derived $CH_4$ emission estimates between the two methodologies.

*Baseline*

When DCG was not implemented, total $N_2O$ and manure-derived $CH_4$ emissions ranged from 1667 to 2656 kg $CO_2$e/ha/year for imperfectly drained soils and from 3015 to 3785 kg $CO_2$e/ha/year for poorly drained soils (Fig. 3). Manure-derived $CH_4$ emissions represented 37-51% and 29-33% of the total $N_2O$ and manure-dived $CH_4$ emissions for the imperfectly and poorly drained soils, respectively. Direct and indirect $N_2O$ emissions from excreta deposition, fertiliser application and manure storage and application (exclusive of $CH_4$ emissions from manure management) ranged from 817 to 1457 kg $CO_2$e/ha/year for imperfectly drained soils, and 2027 to 2552 kg $CO_2$e/ha/year for poorly drained soils (Fig. 3).

For imperfectly drained soils in Manawatu and Canterbury, the largest contribution to direct $N_2O$ emissions was from N fertiliser (40% and 35% of total $N_2O$ emissions, respectively). Whereas, for farms on imperfectly drained soils in Waikato and Southland and on poorly drained soils in all regions, urine deposited directly onto pasture was the largest $N_2O$ source accounting for between 32% and 67% of total $N_2O$ emissions. We explored cross-regional relationships by utilising modelling results from all three modelled years ($10^{th}$, $50^{th}$ and $90^{th}$ percentile rainfall years). Using results from the baseline farms (i.e. DCG not implemented) on two contrasting soil drainage classes in four regions, we observed a significant linear relationship between the number of days VWC was above the CWC threshold (i.e. increasing number of 'wet' days) and total $N_2O$ and manure-derived

$CH_4$ emissions on a per cow per day basis (normalised across regions for differences in stocking rates and days on farm, $R^2$= 0.59, P < 0.001, n=24; Fig. 4).

*Restricted grazing scenarios*

Adopting DCG for 0 hours per day (i.e. complete removal) on farms with poorly drained soils reduced total $N_2O$ and manure-derived $CH_4$ emissions by 4 - 12% in Waikato, Manawatu and Southland (Fig. 3). The reduction in $N_2O$ emissions from urine and dung deposition due to cows being completely removed from wet paddocks was only partially offset by increased $N_2O$ emissions from effluent and manure application and $CH_4$ emissions from manure management. Adopting DCG for 13 or 17 hours per day did not result in the same decline in GHG emissions compared to complete removal of cows, with reductions of between 3 - 9% predicted. In contrast, the Canterbury farms showed little change (0 - +2%) in emissions when DCG was implemented (Fig. 3) due to the drier climate (Table 1). The relative impact of DCG when soils were wet on reducing total $N_2O$ and manure-derived $CH_4$ emissions compared to the baseline varied across regions and increased with the number of 'wet' days. Consequently, DCG was only effective at reducing total $N_2O$ and manure-derived $CH_4$ emissions on poorly drained soils that had more than ca. 150 'wet' days per year (Fig. 5; includes data from the $10^{th}$, $50^{th}$ and $90^{th}$ percentile rainfall years).

*Insert Figure 3*

*Insert Figure 4*

*Insert Figure 5*

For dairy farms with imperfectly-drained soils, complete removal of cows from wet paddocks in Waikato and Manawatu increased total $N_2O$ and manure-derived $CH_4$ emissions by 6-10% (Fig. 3). This reflects an increase in emissions from manures that more than offset the predicted reductions in paddock-based emissions, indicating pollution swapping. Adopting DCG for 13 or 17 hours on wet days had little effect on total $N_2O$ and manure-derived $CH_4$ emissions. In Canterbury and Southland, where cows were wintered off in June and July, there was a small increase of 2 - 4% in the total $N_2O$ and manure-derived $CH_4$ emissions when cows were completely removed from wet paddocks, with very little change (0 - 1%) when DCG was implemented for 13 or 17 hours per day.

3.3 Inventory methodology

The benefits in reduced GHG emissions achieved from adopting DCG were not apparent when emissions were calculated using the current New Zealand inventory methodology. Firstly, estimated total $N_2O$ and manure-derived $CH_4$ emissions for farms on imperfectly drained soils were 30-50% greater compared to the refined method (Fig. 6) primarily due to higher paddock-derived $N_2O$ emissions based on a single $EF_3$ value of 1% for urine compared to lower emissions for imperfectly drained soils based on the VWC and natural logarithic $EF_3$ relationship (Fig. 1). Secondly, the current New Zealand inventory methodology assumes 100% of excreta deposited on standoff pads would be stored in 'anaerobic lagoons' i.e. effluent pond (Table 2), generating large emissions of $CH_4$ (0.1095 kg $CH_4$/kg faecal dry matter). In contrast, the refined method assumes most of the excreta is stored as solid manure (Luo et al., 2008), emitting lower rates of $CH_4$ similar to dung deposition onto pasture (ca. 0.0009 kg $CH_4$/kg faecal dry matter; Table 2; IPCC, 2006).

*Insert Figure 6*

3.5 Cost-benefit of adopting DCG when soils were wet to mitigate GHG emissions

The cost-benefit of our DCG approach ($/t $CO_2$e reduced; Table 4) was calculated for farms on poorly drained soils using modelled total $N_2O$ and manure-derived $CH_4$ emissions based on the refined inventory approach (Fig. 3) and operating profit (Table 3; sourced from Laurenson et al., submitted). We did not include imperfectly drained soils because there was no reduction in GHG emissions when adopting DCG. For poorly drained soils, the cost:benefit of implementing DCG for 13 hours on wet days in Waikato, Manawatu and Southland ranged from a benefit of \$500 per t $CO_2$e reduced (Manawatu) to a cost of \$620 per t of $CO_2$e reduced (Waikato) (Table 4), with higher costs when adopting a longer DCG policy. In contrast to 13 and 17 hour DCG, the cost of completely removing cows from wet paddocks was much greater, at between \$6730 and \$19,000 per t $CO_2$e reduced. In Canterbury, the small reduction in total GHG emissions for the 13 and 17 hour DCG scenarios substantially increased the cost of adoption (\$14,000-15,000 per t $CO_2$e reduced; Table 4). The increase in GHG emissions when cows were completely removed from wet paddocks precluded any benefit of this practice, reflecting the relatively low number of wet days in the Canterbury region and the increase in GHG emissions from manure management (Fig. 3).

*Insert Table 4*

# 4 Discussion

## 4.1 Method of calculation

Our results suggest no benefit can be determined from the proposed DCG for reducing total $N_2O$ and manure-derived $CH_4$ emissions from dairy farms on either imperfectly drained or poorly drained soils when estimated using the current inventory methodology. Adopting a single $EF_3$ value of 1% for urine deposited onto soil ignores the influence of soil wetness (and therefore aeration) on microbial-mediated $N_2O$ production (van der Weerden et al., 2012). The refined approach, where urine $EF_3$ is a function of soil water content, a proxy for soil aeration status, provides a more accurate assessment of the impact of urine deposition on $N_2O$ emissions from wet soils. Another key difference between the two approaches is that the current inventory method assumes any excreta deposited off-paddock is stored in anaerobic lagoons (Ministry for the Environment, 2015), which emit $CH_4$ at rates much greater than for solid manure (IPCC, 2006). This could inflate the accounting of GHG emissions for farms utilising standoff pads. In practice, excreta deposited onto standoff pads is typically stored as a solid material prior to land application, with negligible amounts of excreta entering ponds. Luo et al. (2008) found that only 4% of the liquid from a standoff pad entered the pond, presumably due to the significant retention of effluent in the woodchip bedding material (Dumont et al., 2012). Inclusion of a second manure management category such as 'solid storage' within the inventory methodology would provide a more accurate accounting of emissions from manure deposited onto standoff pads.

## 4.2 Reduction in total $N_2O$ and manure-derived $CH_4$ emissions

The aim of the study was to test if DCG based on a soil water content threshold could reduce farm scale GHG emissions. For poorly drained soils, our DCG approach substantially reduced direct $N_2O$ emissions from excreta deposition when modelled using the refined inventory methodology. The reduction was more than sufficient to offset any increase in $N_2O$ emissions from storage and land application of solid manure. The DCG was most effective at reducing total $N_2O$ emission when cows were completely removed from poorly drained, wet paddocks. In contrast, there was little if any benefit in removing cows from imperfectly-drained soils because the reduction in paddock-based emissions was insufficient to offset a large increase in $N_2O$ emissions associated with storage and land application of solid manure.

When including manure-derived $CH_4$ emissions, implementation of DCG for imperfectly drained soils at the threshold tested will lead to an increase in GHG emissions. Whereas, the CWC used for poorly drained soils led to substantial reductions in total emissions when DCG was implemented, particularly when there are more than 150 'wet days' per year (i.e. VWC > CWC).

Previous studies proposed implementation of DCG practices during 'high risk' periods such as autumn/winter i.e. a calendar approach (de Klein et al., 2006; Luo et al., 2013) in contrast to our tactical approach. On a poorly drained soil in Southland, limiting cow grazing time to 3 hours per day in autumn (cows wintered off farm for 3 months) reduced total (direct and indirect) on-farm $N_2O$ emissions by 7-11% (de Klein et al., 2006). However, no provision of standoff was made for when soils were wet. Our study showed, for the same region yet cows were wintered off-farm for 2 months only, restricting grazing to 13 hours on wet days reduced $N_2O$ emissions by 9-17% (range of wet, dry and 20-year average rainfall; data not shown). Essentially, our DCG approach produced a greater reduction in total $N_2O$ emissions with less time removed from paddocks compared to de Klein et al.'s (2006) calendar approach. It is also important to note that the earlier study adopted the inventory methodology when modelling $N_2O$ emissions from storage and land application of effluent.

Beukes et al. (2011), using the WFM, also adopted a calendar approach when modelling the effectiveness of standoffs as one of five different on-farm GHG mitigation options in the Waikato. They modelled standoff use at 12 hours per day for two months in autumn (March and April) on a dairy farm on a well-drained soil. Total GHG emissions (which included $CH_4$ enteric fermentation) did not decrease because the reduced $N_2O$ emission from urinary N deposited onto pasture was fully offset by GHG emissions associated with the standoff pad and the application of manure onto pasture. In our study, the Waikato results for an imperfectly drained soil also showed no net decline in total $N_2O$ and manure-derived $CH_4$ emissions, even though we used soil moisture to derive an $EF_3$ value and a CWC to remove cows from wet soils. Essentially, on soils that have reasonably good drainage and therefore relatively low $N_2O$ emissions, removing cows from wet soils is likely to result in pollution swapping.

Removing cows from wet paddocks will also reduce N leaching, which, in addition to being an indirect source of $N_2O$ emissions (Fig. 3), is a water quality pollutant of major concern in New Zealand (de Klein et al., 2006; Christensen et al., 2012). Our modelled data suggests implementing a 13 hour per day DCG policy could reduce N leaching by up to 13%, providing a co-benefit for its use (data not shown). However, this is a smaller reduction than when compared to complete removal of cows from paddocks during autumn months (following a calendar approach), resulting in ca. 40% reduction in N leaching (de Klein et al., 2006; Vogeler et al., 2013). Therefore, the use of off-paddock facilities will ultimately be dependent on the goals farmers are trying to achieve.

### 4.3 Cost effectiveness of DCG based on a soil water threshold

Our analysis suggests adopting a DCG of 13 hours per day on farms with poorly drained soils when soils are wet is most cost-effective in terms of reducing GHG emissions, particularly if the number of 'wet' days per year is greater than 150 days. Recently, Vibart et al. (2015) assessed the cost:benefit of a package of mitigation options for Southland dairy farms, which included DCG in addition to other changes including construction of a covered loafing pad and installation of a low rate effluent application. On a dairy farm system similar to that modelled for Southland, this mitigation package cost \$940/t $CO_2$e reduced relative to the baseline. However, it is difficult to single out the influence of the DCG practice on this value. A more recent analysis showed that employing a standoff for 8 hours per day in March and April, with 50% of the herd on a loafing pad in May and June, resulted in a cost of \$2600/t $CO_2$e reduced (R. Vibart, unpubl. data). Both Vibart's studies used the OVERSEER® model to calculate the GHG emissions (Wheeler et al., 2008), where $N_2O$ emissions from excreta deposition increase with increasing soil water content. While both our refined approach and the OVERSEER predicts urine-derived $N_2O$ emissions in response to soil water content, the former is sensitive to daily changes in soil water content. In contrast, the OVERSEER model operates on a coarser monthly time-step and is therefore less sensitive to rainfall and irrigation events. Adler et al. (2015) used the WFM and New Zealand-specific emission factors to analyse the cost of GHG mitigation strategies for dairy farms in Waikato and Canterbury and found that off-paddock facilities such as standoff pads were a costly alternative compared to other mitigation options such as lower stocking rates and reduced N fertiliser use.

Although implementing DCG at 13 hours on wet days was most cost effective for poorly drained soils, the cost:benefit values ranged widely between regional climates, from a desirable benefit of \$500/ t $CO_2$e reduced in the Manawatu to a cost of \$540-\$620/t $CO_2$e reduced in Waikato and Southland. The negligible reduction in modelled GHG emissions in Canterbury made DCG financially unviable (estimated cost of \$14,000/t $CO_2$e reduced). In regions where cows are removed from the dairy platform over the winter months (i.e. 'wintered off') such as Canterbury and Southland, the impact of DCG on reducing farm-scale GHG emissions will be limited compared to many North Island regions. This will impact on the financial viability of installing off-paddock facilities such as standoffs with the purpose of reducing GHG emissions due to their associated low return on investment (Adler et al., 2015; Laurenson et al., submitted). Our financial analysis included capital costs associated with construction of the off-paddock facility; our proposed DCG approach will be more financially attractive for farms where off-paddock facilities already exist. It should be noted that our analysis assumed a long-term milk payout of \$6/kg MS (Laurenson et al., submitted).

In the current study it was assumed farms were located on a single soil type: future modelling should include farms with mixed soil types (drainage classes). Also, more information on the impact of treading damage and subsequent pasture production, and interaction between damaged soil and urine/dung deposition on $N_2O$ emissions or EFs is needed. Improved understanding of how soil aeration status, relative diffusivity and appropriate methods for measuring or estimating how these parameters affect $N_2O$ emissions is required. This will assist with improving relationships for estimating the impact of grazing and soil damage on emission factors for excreta, fertiliser and manure application to soils.

## 5 Conclusions

Our analysis suggests that, on farms with poorly drained soils, limiting grazing time to 13 hours per day when soils are wet is most cost-effective when aiming to reduce total $N_2O$ and manure-derived $CH_4$ emissions, particularly if the number of 'wet' days (i.e. VWC > CWC) is greater than 150 days. In contrast, there was an increase in emissions for dairy farms on imperfectly-drained soils, suggesting our proposed DCG approach is not suitable for reducing GHG emissions on these soils.

## 6 Acknowledgements

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Figure Captions

Figure 1. Relationship between modelled VWC averaged over 20 days from time of urine deposition and natural log of measured dairy cattle urine $N_2O$ emission factor (ln $EF_3$, %)

Figure 2. Comparison of direct $N_2O$ emissions (kg $N_2O$-N/ha/yr) from urine deposition on grazed paddocks when adopting DCG for 0 (i.e. complete removal), 13 or 17 hours per day compared to 21 hours per day (baseline) when soil moisture > CWC for an imperfectly-drained (●) and poorly drained (□) soil, calculated using a refined methodology based on soil moisture content (left) and the current New Zealand inventory methodology (right). Values modelled for the 50$^{th}$ percentile rainfall year**.**

Figure 3. Total $N_2O$ and manure-derived $CH_4$ emissions ('Total emissions', kg $CO_2$e/ha/year) from baselines and 3 duration controlled grazing scenarios (0 (i.e. complete removal), 13 and 17 hours' grazing per day when soil moisture > CWC) for an imperfectly-drained and poorly drained soil in four regions (a: Waikato, b: Manawatu, c: Canterbury, d: Southland). Values modelled for the 50$^{th}$ percentile rainfall year using a refined inventory methodology**.**

Figure 4: Relationship between number of days VWC > CWC and total $N_2O$ and manure-derived $CH_4$ emissions (kg $CO_2$e/cow/day) for baseline (i.e. cows not removed from wet paddocks). Values modelled for two drainage classes by four regions by three years (10$^{th}$, 50$^{th}$ and 90$^{th}$ percentile rainfall years) (n=24) using a refined inventory methodology**.**

Figure 5: Relationship between number of days VWC > CWC and reduction in total $N_2O$ and manure-derived $CH_4$ emissions (kg $CO_2$e/cow/day) when duration controlled grazing implemented for 0, 13 or 17 hours per day for poorly drained soils only. Values modelled for two drainage classes by four regions by three years ($10^{th}$, $50^{th}$ and $90^{th}$ percentile rainfall years) (n=24) using a refined inventory methodology**.**

Figure 6: Comparison of total $N_2O$ and manure-derived $CH_4$ emissions ('Total emissions', kg $CO_2$e/ha/year) based on current New Zealand inventory methodology (□) and refined methodology (■) from baseline dairy farms and when DCG implemented for 0 (i.e. no grazing), 13 and 17 hours per day when soil moisture > CWC on an imperfectly-drained and poorly drained soil in four regions (a: Waikato, b: Manawatu, c: Canterbury, d: Southland). Values modelled for the $50^{th}$ percentile rainfall year. Black bars correspond to the total emissions reported in Fig 3.

Table 1. Details of regions, climates and dairy farm production values.

| Region | Location | Coordi-nates | Year[A] | Relative Rainfall[A] | Actual Rainfall (mm) | Typical pasture production (t DM/ha/yr) | | Stocking rate (cows/ha) | | No. days above CWC[B] | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | Imperfectly drained | Poorly drained | Imperfectly drained | Poorly drained | Imperfectly drained | Poorly drained |
| Waikato | Hamilton | 37.775 S, 175.325 E | 2013-14 | 10th percentile | 873 | 12.0 | 12.0 | 2.95 | 2.95 | 83 | 201 |
| | | | 2012-13 | 50th percentile | 1097 | 14.5 | 14.0 | | | 111 | 181 |
| | | | 2010-11 | 90th percentile | 1439 | 17.5 | 17.0 | | | 141 | 243 |
| Manawatu | Palmerston North | 40.375 S, 175.625 E | 2007-08 | 10th percentile | 845 | 10.0 | 9.0 | 2.95 | 2.95 | 87 | 204 |
| | | | 1996-97 | 50th percentile | 1000 | 12.8 | 11.3 | | | 102 | 213 |
| | | | 1995-96 | 90th percentile | 1220 | 14.0 | 11.3 | | | 156 | 270 |
| Canterbury | Lincoln | 43.625 S, 172.475 E | 1998-99 | 10th percentile | 471 (+375 [C]) | 17.0 | 17.0 | 3.9 | 3.9 | 15 [D] | 142 [D] |
| | | | 2007-08 | 50th percentile | 631 (+325 [C]) | 18.5 | 18.5 | | | 25[D] | 152 [D] |
| | | | 2008-09 | 90th percentile | 879 (+275 [C]) | 20.0 | 20.0 | | | 41 [D] | 147 [D] |
| Southland | Winton | 46.125 S, 168.325 E | 2002-03 | 10th percentile | 823 | 9.7 | 15.1 | 2.75 | 3.15 | 50 [D] | 135 [D] |
| | | | 1999-2000 | 50th percentile | 898 | 11.0 | 17.4 | | | 32 [D] | 145 [D] |
| | | | 1996-97 | 90th percentile | 1017 | 12.5 | 20.0 | | | 71 [D] | 172 [D] |

[A] Year was chosen based on the 10th, 50th and 90th percentile rainfall experienced in each region between 1995 and 2014; [B] CWC = critical water content; [C] Values in brackets refer to irrigation applied (mm) to supplement rainfall (applied when soil water deficit of 20-25 mm present), [D] Excludes June, July and early August, when cows were wintered off farm.

Table 2: Calculation of total greenhouse gas emissions (excluding enteric fermentation) for modelled dairy farms using New Zealand IPCC inventory methodology and improvements to methodology.

| Component of calculation | Code | New Zealand IPCC inventory methodology | Potential improvements to inventory methodology | Comments |
|---|---|---|---|---|
| $N_2O$ emission factor for urine (kg $N_2O$-N/kg N) | $EF_{3PRP}$ | 0.01 | Dependent on soil water content. | Based on relationship between $EF_{3PRP}$ and VWC (Fig. 1). |
| $N_2O$ emission factor for dung (kg $N_2O$-N/kg N) | $EF_{3PRP\ DUNG}$ | 0.0025 | NC[A] | |
| $N_2O$ emission factor for urea fertiliser (kg $N_2O$-N/kg N) | $EF_{1\ UREA}$ | 0.0048 | 0.006 | van der Weerden et al. (2016) |
| Fraction of $N_{EX}$ or urea fertiliser N leached (kg $NO_3$-N/kg N) | $Frac_{LEACH}$ | 0.07 | Modelled using APSIM | Uses local climate and soil data |
| Fraction of FDE N leached (kg $NO_3$-N/kg N) | $Frac_{LEACH\ FDE}$ | 0.07 | NC | |
| $N_2O$ emission factor for N leached (kg $N_2O$-N/kg N) | $EF_5$ | 0.0075 | NC | |
| Fraction of $N_{EX\ URINE}$ lost through $NH_3$ volatilisation (kg $NH_3$-N/kg N) | $Frac_{GASM\ URINE}$ | 0.10 | Modelled using APSIM | Uses local climate and soil data |
| Fraction of $N_{EX\ DUNG}$ lost through $NH_3$ volatilisation (kg $NH_3$-N/kg N) | $Frac_{GASM\ DUNG}$ | 0.10 | NC | |
| Fraction of urea fertiliser lost through $NH_3$ volatilisation (kg $NH_3$-N/kg N) | $Frac_{GASF}$ | 0.10 | Modelled using APSIM | Uses local climate and soil data |
| $N_2O$ emission factor for $NH_3$ volatilisation (kg $N_2O$-N/kg N) | $EF_4$ | 0.01 | NC | |
| $N_2O$ emission factor effluent storage in uncovered anaerobic lagoon (kg $N_2O$-N/kg N) | $EF_{3(S\ AL)}$ | 0 | NC | |
| $N_2O$ emission factor excreta deposited onto standoff pad (=solid storage). (kg $N_2O$-N/kg N) | $EF_{3(S\ SS)}$ | Not considered; therefore treated all excreta on standoff pad as $EF_{3(S\ AL)}$ (= 0) | 0.0001 | Luo and Saggar (2008) |
| Fraction of effluent N leached during storage in uncovered anaerobic lagoon (kg $NO_3$-N/kg N) | $Frac_{LEACH\ MS}$ | 0 | NC | |

| | | | | |
|---|---|---|---|---|
| Fraction of effluent N lost as $NH_3$ during storage (kg $NH_3$-N/kg N) | $Frac_{GasMS\ FDE}$ | 0.35 | NC | |
| Fraction of stored effluent in anaerobic lagoon lost during storage as gaseous N (kg N/kg N) | $Frac_{LossMS\ FDE}$ | 0.35[B] | NC | 2006 IPCC guidelines, Chapter 10, Table 10.23 (IPCC, 2006) |
| Fraction of stored effluent applied to land, adjusted for N lost during manure management system (kg N/kg N) | $FracN_{EX\ EFFAPP}$ | 1 - $Frac_{LossMS\ FDE}$ = 0.65 | NC | 2006 IPCC guidelines, Chapter 10, Equation 10.34 (IPCC, 2006) |
| Fraction of excreta N from standoff pad MM lost as $NH_3$ (kg $NH_3$-N/kg N) | $Frac_{GasMS\ SO}$ | Not considered; therefore treated as effluent (0.35) | 0.30 | Assumed Standoff pad = 'Solid Storage' MM (2006 IPCC guidelines, Chapter 10, Table 10.18); $Frac_{GasMS}$ given in Table 10.22 (IPCC, 2006) |
| Fraction of standoff pad N excreta entering anaerobic lagoon (kg N/kg N) | $FracN_{EX\ SO \rightarrow AL}$ | Not considered; therefore treated as effluent (1.0) | Assumed 0.04 | Luo et al. (2008) |
| Fraction of standoff pad excreta applied to land, adjusted for N lost during manure management system (kg N/kg N) | $FracN_{EX\ SO\ APP}$ | Not considered; therefore assumed same as FDE: 1 - $Frac_{LossMS\ FDE}$ = 0.65 | Assume 1 – ($EF_{3\ S\ SS}$ + $Frac_{LossMS\ SO}$) = 0.70 | |
| $N_2O$ emission factor for farm dairy effluent (kg $N_2O$-N/kg N) | $EF_{1\ FDE}$ | 0.01 | 0.003 | van der Weerden et al. (2016) |
| $N_2O$ emission factor for standoff pad manure applied to land (kg $N_2O$-N/kg N) | $EF_{1\ SO}$ | 0.01 | NC | |
| Fraction of $N_{EX}$, applied FDE or applied standoff pad manure lost through $NH_3$ volatilisation (kg $NH_3$-N/kg N) | $Frac_{GASM}$ | 0.10 | NC | |
| Fraction of standoff pad manure N leached (kg $NO_3$-N/kg N) | $Frac_{LEACH\ SO}$ | Not considered; therefore treated as effluent (0.07) (consistent with other N loss pathways) | 0 | Assumed Standoff pad manure applied to land under good practice. Also, less mobile form of N. |
| $CH_4$ emissions for $N_{EX\ DUNG}$ deposited onto pasture (kg $CH_4$/kg FDM[C]) | $CH_{4\ PRP}$ | 0.00098 | NC | |
| $CH_4$ emissions from stored effluent in | $CH_{4\ MM}$ | 0.1095 | NC | |

| | | | | |
|---|---|---|---|---|
| anaerobic lagoons (kg $CH_4$/kg FDM stored) | | | | |
| $CH_4$ emissions from excreta deposited onto standoff pads (= solid storage) (kg $CH_4$/cow/year) | $CH_{4\,MM}$ | Not considered, therefore treated as effluent FDM entering anaerobic lagoons. | VS = 3.5 kg/cow/day; $B_O$ = 0.24; MCF = 4% (see footnote for description). | 2006 IPCC guidelines, Chapter 10, Equation 10.23; Table 10.A4 (IPCC, 2006) |

[A] NC = no change to NZ inventory methodology; [B] NZ inventory assumes all gaseous N losses from anaerobic lagoon are as $NH_3$, with nil $N_2$ emissions (MPI, pers. comm.2015); [C] FDM = Faecal dry matter; VS = volatile solids; $B_O$ = maximum methane producing capacity for manure produced by cattle; MCF = methane conversion factor.

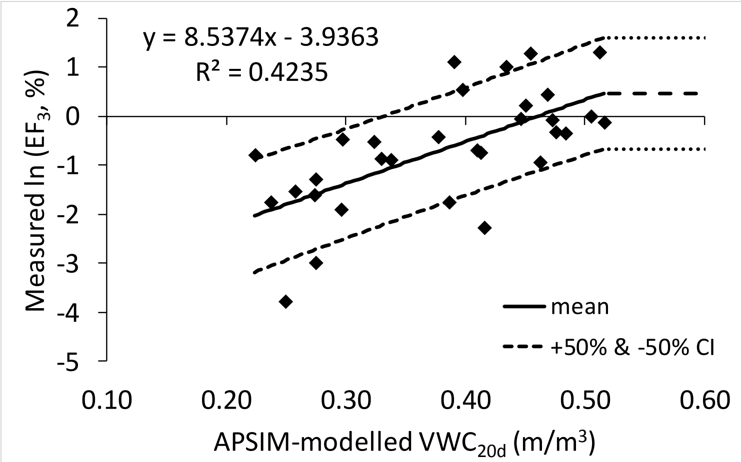
Table 3. Change in dairy operating profit (DOP; $/ha/year) for duration controlled grazing (DCG) scenarios (0, 13 and 17 hours grazing per day when soils are wet) for dairy farms with poorly drained soils. Negative values indicate a reduction in DOP, positive values indicate an increase in DOP. Values shown are for the 50th percentile rainfall year (source: Laurenson et al. submitted.).

| DCG (hours/day when VWC > CWC) | Region | | | |
|---|---|---|---|---|
| | Waikato | Manawatu | Canterbury | Southland |
| 0 (no grazing) | -$2,539 | -$2,299 | -$1,843 | -$1,522 |
| 13 | -$148 | +$81 | -$155 | -$142 |
| 17 | -$222 | -$91 | -$178 | -$139 |

Table 4. Cost:benefit of contrasting duration controlled grazing (DCG) scenarios when poorly drained soils were wet to reduce GHG emissions ($ per t $CO_2$e reduced). Negative values indicate a cost, positive values indicate a benefit; for Canterbury, 0 hours excluded due to emissions increasing relative to baseline. Values shown are based on 20-year average rainfall.

| DCG scenarios (hours/day when VWC > CWC) | Region | | | |
|---|---|---|---|---|
| | Waikato | Manawatu | Canterbury | Southland |
| 0 (no grazing) | -$6,730 | -$19,000 | | -$7,320 |
| 13 | -$620 | +$500 | -$14,130 | -$540 |
| 17 | -$1,520 | -$900 | -$14,800 | -$1,340 |

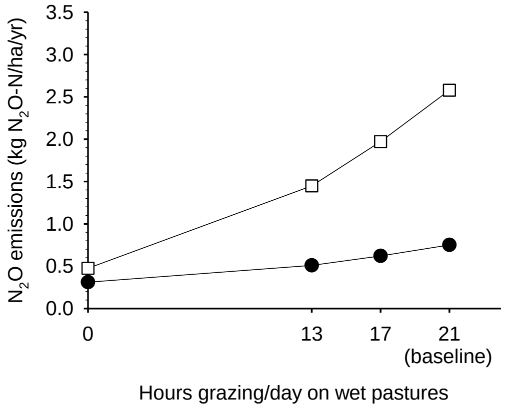
Fig. 1.



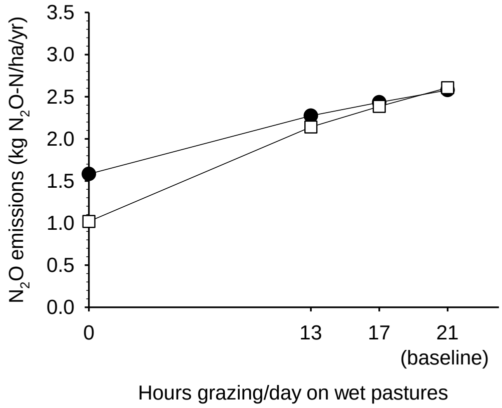
y = 8.5374x - 3.9363
R² = 0.4235
Measured ln (EF3, %)
APSIM-modelled VWC20d (m/m3)
mean
+50% & -50% CI
2
1
0
-1
-2
-3
-4
-5
0.10
0.20
0.30
0.40
0.50
0.60

Fig. 2.

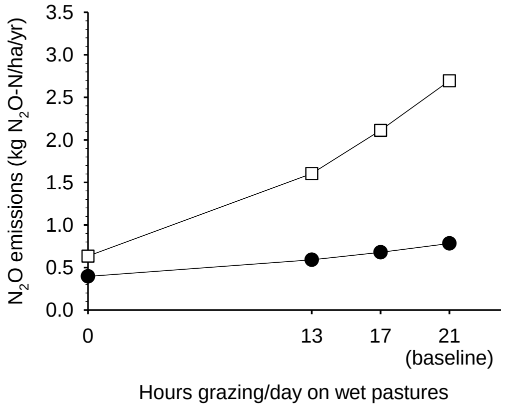
(a) Waikato - Refined



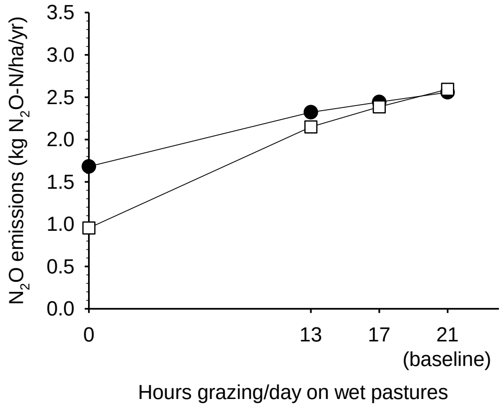

(b) Waikato – Inventory



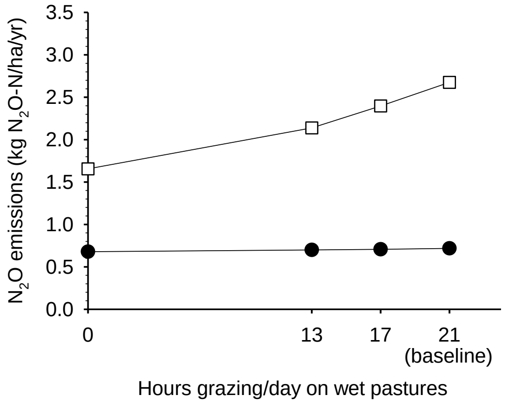

(c) Manawatu – Refined



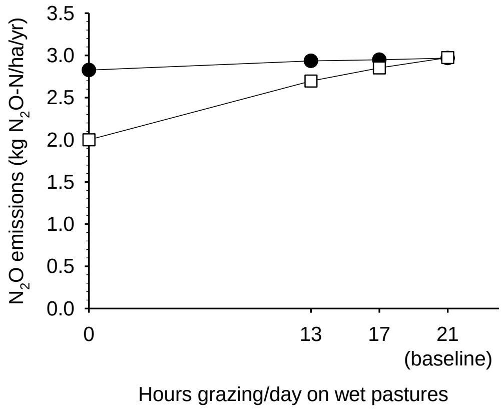

(d) Manawatu – Inventory




(e) Canterbury – Refined

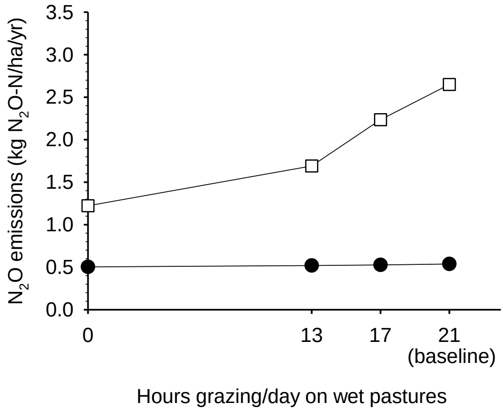



(f) Canterbury – Inventory

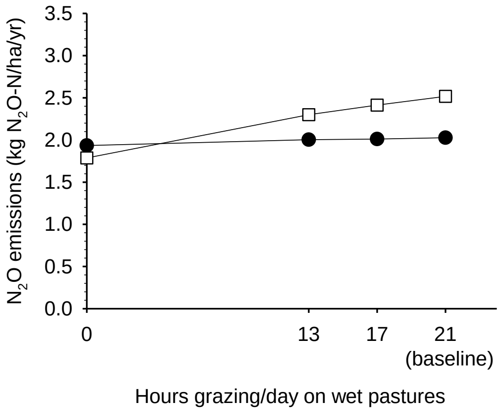



(g) Southland – Refined

(h) Southland – Inventory

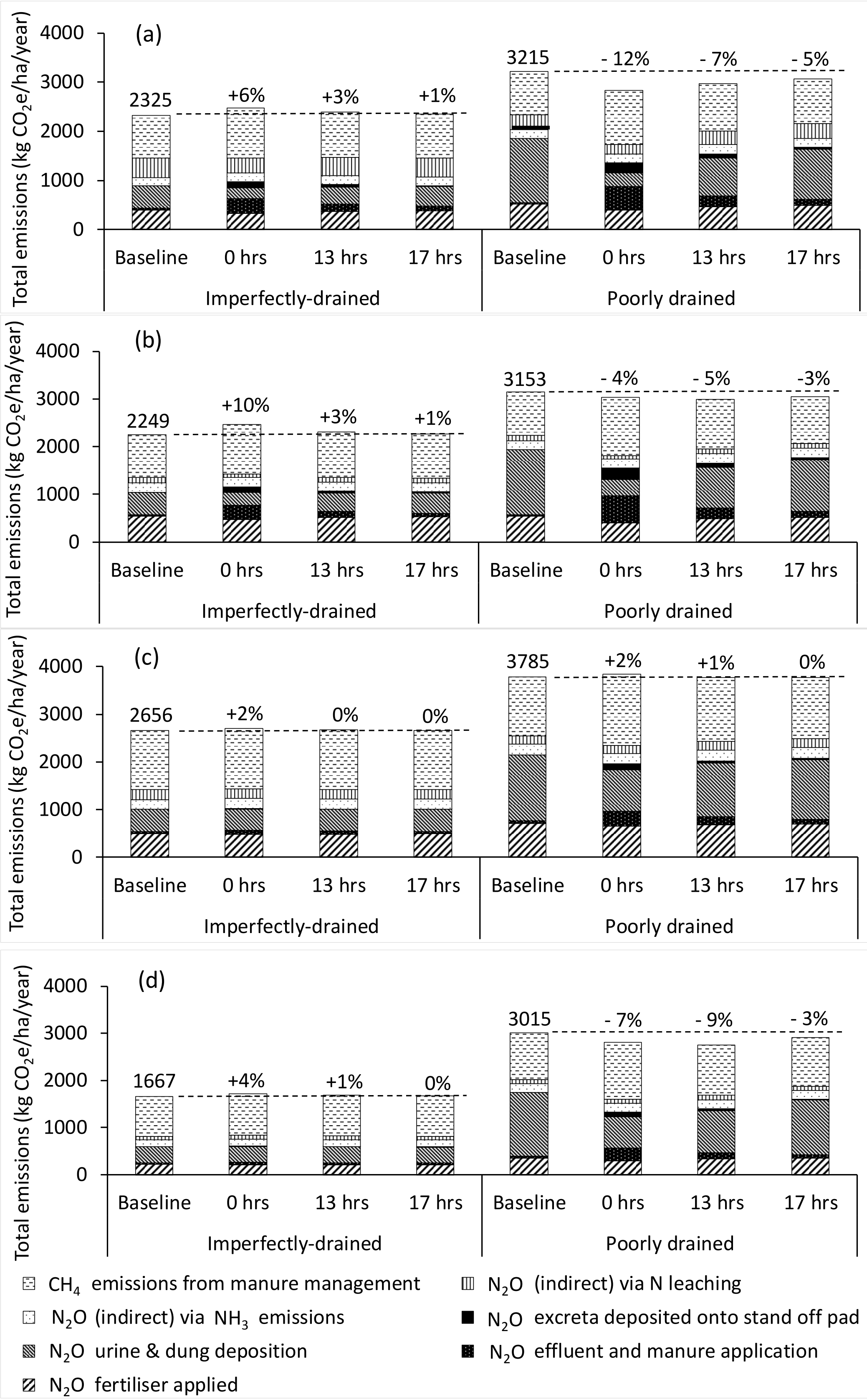


N2O emissions (kg N2O-N/ha/yr)
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
0
13
17
21
(baseline)
Hours grazing/day on wet pastures

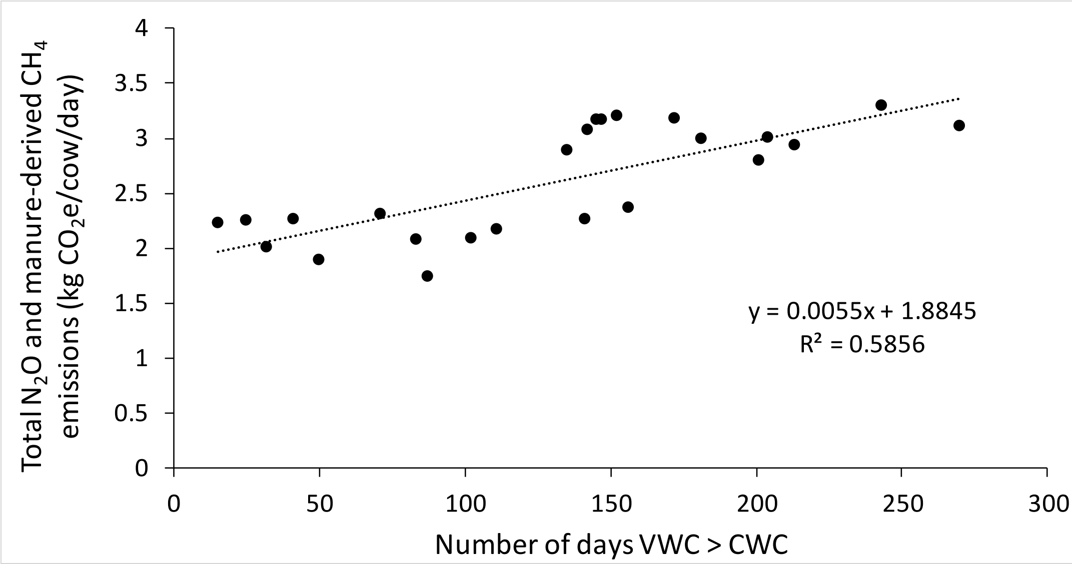
N2O emissions (kg N2O-N/ha/yr)
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
0
13
17
21
(baseline)
Hours grazing/day on wet pastures

Fig. 3.



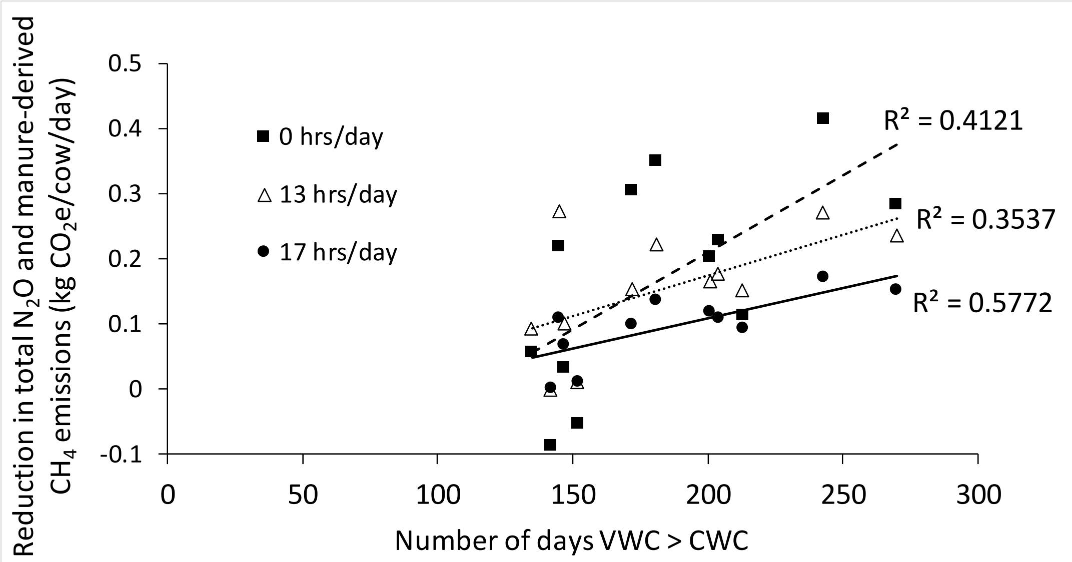
(a)
(b)
(c)
(d)
Total emissions (kg $CO_2$e/ha/year)
4000
3000
2000
1000
0
2325
+6%
+3%
+1%
3215
- 12%
- 7%
- 5%
2249
+10%
+3%
+1%
3153
- 4%
- 5%
-3%
2656
+2%
0%
0%
3785
+2%
+1%
0%
1667
+4%
+1%
0%
3015
- 7%
- 9%
- 3%
Baseline
0 hrs
13 hrs
17 hrs
Imperfectly-drained
Poorly drained
$CH_4$ emissions from manure management
$N_2O$ (indirect) via N leaching
$N_2O$ (indirect) via $NH_3$ emissions
$N_2O$ excreta deposited onto stand off pad
$N_2O$ urine & dung deposition
$N_2O$ effluent and manure application
$N_2O$ fertiliser applied

Fig. 4.



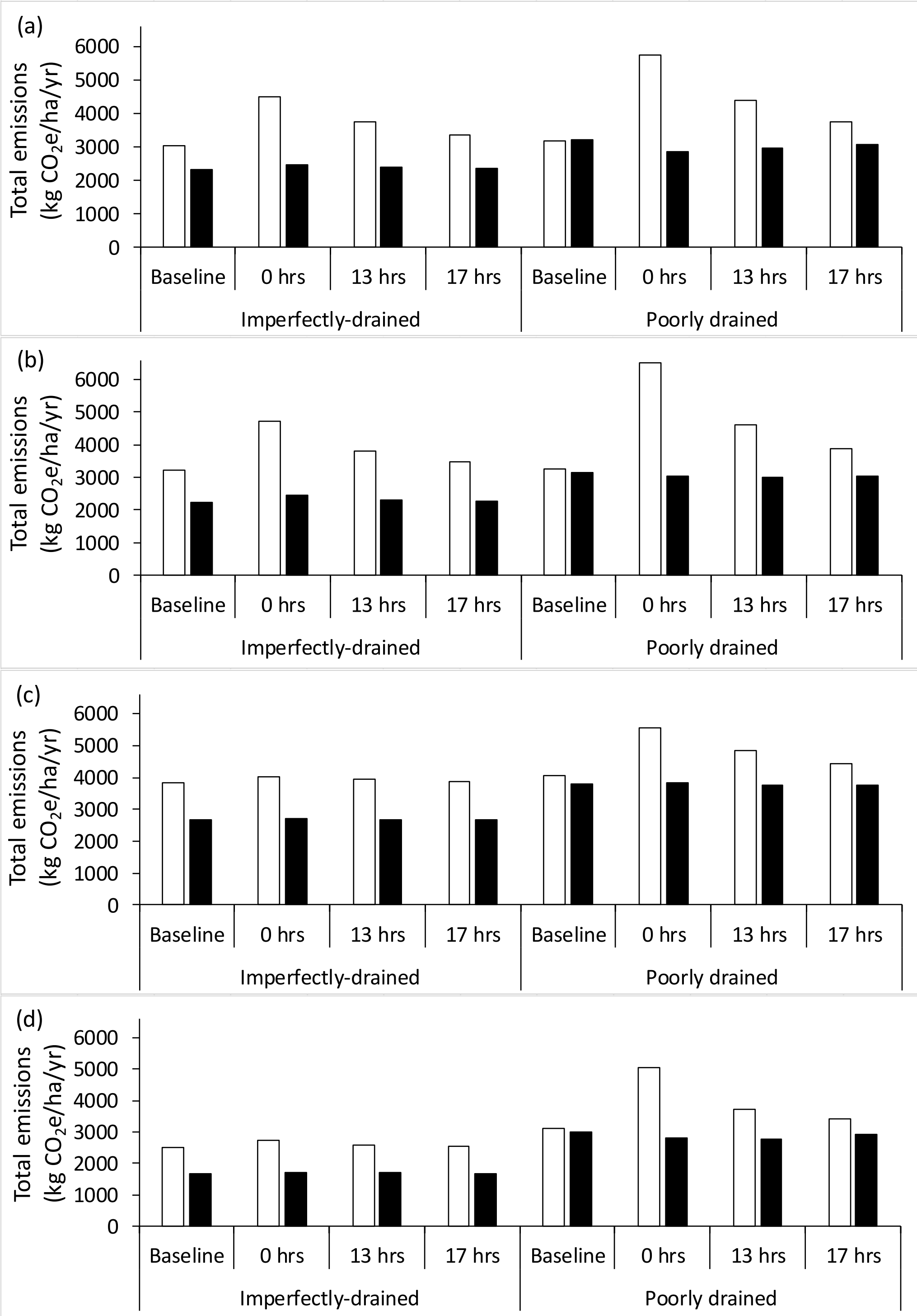
Total $N_2O$ and manure-derived $CH_4$ emissions (kg $CO_2$e/cow/day)
4
3.5
3
2.5
2
1.5
1
0.5
0
0
50
100
150
200
250
300
Number of days VWC > CWC
y = 0.0055x + 1.8845
$R^2$ = 0.5856

Fig. 5.



Reduction in total $N_2O$ and manure-derived $CH_4$ emissions (kg $CO_2$e/cow/day)
0.5
0.4
0.3
0.2
0.1
0
-0.1
0 hrs/day
13 hrs/day
17 hrs/day
$R^2$ = 0.4121
$R^2$ = 0.3537
$R^2$ = 0.5772
0
50
100
150
200
250
300
Number of days VWC > CWC

Fig. 6.


(a)
Total emissions (kg CO2e/ha/yr)
6000
5000
4000
3000
2000
1000
0
Baseline
0 hrs
13 hrs
17 hrs
Baseline
0 hrs
13 hrs
17 hrs
Imperfectly-drained
Poorly drained
(b)
(c)
(d)